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UNCERTAINTIES IN LCA

Application of three independent consequential LCA approaches to the agricultural sector in Luxembourg

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Abstract
Purpose Consequential Life Cycle Assessment (C-LCA) is a "system modelling approach in which activities in a product system are linked so that activities are included in the product system to the extent that they are expected to change as a consequence of a change in demand". Hence, C-LCA focuses on micro-economic actions linked to macro-economic consequences, by identifying the (marginal) suppliers and technologies prone to be affected by variable scale changes in the demand of a product. Detecting the direct and indirect environmental effects due to changes in the production system is not an easy task. Hence, researchers have combined the consequential perspective with different econometric models. Therefore, the aim of this study is to assess an increase in biocrops cultivation in Luxembourg using three different consequential modelling approaches to understand the benefits, drawbacks and assumptions linked to each approach as applied to the case study selected.
Methods Firstly, a partial equilibrium (PE) model is used to detect changes in land cultivation based on the farmers' revenue maximisation. Secondly, another PE model is proposed, which considers a different perspective aiming at minimising a total adaptation cost (so-called opportunity cost) to satisfy a given new demand of domestically produced biofuel. Finally, the *consequential system delimitation for agricultural LCA* approach, as proposed by Schmidt (Int J Life Cycle Assess 13:350–364, 2008), is applied.
Results and discussion The two PE models present complex shifts in crop rotation land use changes (LUCs), linked to the optimisation that is performed, while the remaining approach has limited consequential impact on changes in crop patterns since the expert opinion decision tree constitutes a simplification of the ongoing LUCs. However, environmental consequences in the latter were considerably higher due to intercontinental trade assumptions recommended by the experts that were not accounted for in the economic models. Environmental variations between the different scenarios due to LUCs vary based on the different expert- or computational-based assumptions. Finally, environmental consequences as compared with the current state-of-the-art are lame due to the limited impact of the shock within the global trade market.
Conclusions The use of several consequential modelling approaches within the same study may help widen the interpretation of the advantages or risks of applying a specific change to a production system. In fact, different models may not only be good alternatives in terms of comparability of scenarios and assumptions, but there may also be room for complementing these within a unique framework to reduce uncertainties in an integrated way.

Keywords Agriculture · Consequential LCA · Expert opinion · Land use changes · Partial equilibrium model · Scenario modelling

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1 Introduction

Life cycle assessment (LCA) has acquired a great level of development when it comes to describing the environmental impacts of products, processes or services in a wide range of economic sectors (Curran 1996; Sánchez et al. 2012). This perspective, named attributional LCA (A-LCA), embraces a broad number of advantages, which include environmental reporting and the understanding of the main carriers of environmental impact within the analysed production system (Brander et al. 2009) under status-quo conditions. However, A-LCA shows important limitations when it comes to supporting policy-making or industrial decision-making, mainly due to the omission of potential indirect effects engendered in the markets by the underlying actions. Therefore, consequential LCA (C-LCA), which is defined as a life cycle modelling approach which pursues a quantification of the environmental consequences of a specific decision (Ekvall 2002; UNEP 2011; Zamagni et al. 2012), is increasingly used to link micro-economic actions to macro-economic consequences, by identifying the (marginal) suppliers and technologies prone to be affected by large scale fluctuations in demand (Dalgaard et al. 2008; Schmidt and Weidema 2008; Hertel et al. 2010). In fact, C-LCA, which is also known as the change-oriented approach, has shown to be a useful methodology to monitor the indirect environmental consequences linked to bioenergy production (Reinhard and Zah 2011; Sánchez et al. 2012), since a consequential perspective accounts for the current worldwide land use changes (LUCs) occurring when one tries to expand the production of energy crops (Searchinger et al. 2008). Indeed, consequential LCA can be defined as "a system modelling approach in which activities in a product system are linked so that activities are included in the product system to the extent that they are expected to change as a consequence of a change in demand for the functional unit" (UNEP 2011).

Detecting marginal technologies or crops subject to suffer indirect effects due to changes in the production system is not an easy endeavour (Ekvall and Weidema 2004; Brander et al. 2009). These issues are of special relevance in agricultural systems, particularly biocrops, due to the displacement of other crops that these will eventually generate, which will propagate throughout the entire global agricultural system until it levels off through intensification, expansion or, more commonly, a mix of the two effects (Bryngelsson and Lindgren 2013; Gelfand et al. 2013; Sánchez et al. 2012; Searchinger et al. 2008; Tonini et al. 2012). In theory, all displacement, intensification and expansion steps should be considered in order to attribute the environmental consequences of an increased cultivation of the energy crop that is being analysed. However, in practice, this approach is not feasible due to the far reaching *domino effect* that these changes would imply (Schmidt 2008). In addition, another constraint when using agricultural outlooks for the identification of marginal crop suppliers is linked to the fact that projected production increases are not necessarily demand-driven, thus hampering the direct correlation between suppliers and changes in demand (Weidema et al. 2009).

Hence, researchers have started to apply a wide range of different models in combination with the consequential life cycle thinking approach to assess the LUCs and related consequences (Kløverpris and Wenzel 2007). Most of these models are of economic nature and are obtained by adapting their principles to the specific characteristics of the analysed system (Dandres et al. 2012). For instance, previous studies suggest the use of dynamic economic models to identify possible consequences occurring in the agricultural stage of a given system (Kløverpris and Wenzel 2007; Kløverpris and Mueller 2013). On the contrary, other perspectives advocate for developing a simplified model of the related effects in C-LCA to obtain manageable systems, based on default assumptions and decision nodes, as well as assuming long-term full market elasticity (Schmidt 2008).

The main goal of this study is to evaluate one single case study related to maize cultivation for biomethane production in Luxembourg using three different consequential modelling approaches, performing a comparative analysis to determine the degree of convergence or divergence of C-LCA results and the underlying reasons, when using the different perspectives. The crop choice is linked to the fact that maize constitutes a commonly employed crop for energy purposes in Luxembourg (Jury et al. 2010; Udelhoven et al. 2013). Firstly, an economic partial equilibrium (PE) model is used to detect changes in land cultivation based on farmers' revenue maximisation. It considers the agricultural sector as a closed system, allowing detailed representations of agricultural and land use restrictions. Secondly, another PE perspective is proposed, to minimise a total adaptation cost to satisfy a new demand of domestically produced biofuel. Finally, the *consequential system delimitation for agricultural LCA* approach, presented by Schmidt (2008), seeking expert criteria to model the economic assumptions, was also modelled. In this study the term "consequential modelling" is applied to all three methodological approaches to evaluate the activities affected by the change in the functional unit (FU), in accordance with the *Shonan* Guidance Principles (UNEP 2011), therefore not restricting the definition of C-LCA to the latter approach (Schmidt 2008) exclusively.

2 Materials and methods

2.1 Goal and scope definition

The selected case study aims at evaluating the environmental changes in the agricultural sector in Luxembourg linked

to an expected increase in maize cultivation for energy generation. This additional input (hereinafter called *shock*), which was set at 80,000 tonnes per year, was estimated based on the 2020 target fixed for biogas production by the Luxembourgish Renewable Energy Action Plan (LUREAP) (Ministère de l'Economie 2010; see section S1 in the Supplementary Material). Results are expected to provide an interesting field for discussion on the suitability of using different consequential modelling approaches, as well as deliver discussion on how model selection may contribute to the uncertainties existing in C-LCA scenarios.

2.2 Assessment method for the models used for C-LCA

A broad set of possibilities are available when it comes to selecting an approach to assess LUCs and their related consequences:

Economy-Wide (Computable general equilibrium) models These provide a complete representation of national economies, together with a specification of trade relations between economies (van Tongeren et al. 2001) and are based on the notion of market clearance. Computable general equilibrium (CGE) is commonly used whenever allocation of production factors over alternative uses is affected by certain policies or exogenous developments. In fact, international trade is typically an area where these induced effects may be important consequences of policy choices. One of the CGE models which is widely used for trade analysis is the Global Trade Analysis Project (GTAP) (Narayanan et al. 2012), which is based on neoclassical economics (Hertel 1997).

Partial Equilibrium (PE) models These models treat international markets for a selected set of traded goods (e.g. agricultural goods) and are driven by optimisation. PE models adapted to agriculture consider the agricultural system as a closed system without linkages to the rest of the economy, although the effects of the rest of the domestic and world economy on the agricultural system may be included in a top-down fashion by altering parameters and exogenous variables. In addition, PE models are designed to provide a great degree of product detail (Kretschmer and Peterson 2010). The main differences between CGE and PE models in terms of their adaptation to agricultural systems are linked to the explicit modelling of the factor market for land, which in the case of PE models is usually driven by reduced-form supply equations. Moreover, CGE models tend to lead to lower increases of agricultural prices, which are related to the large implicit supply elasticities of these models (Kretschmer and Peterson 2010). Finally, some authors argue that results reported by CGE models usually provide a better picture in the medium-term, while PE models are designed for short-term changes (Ghallagher 2008).

Consequential system delimitation for agricultural LCA approach An alternative to the use of economic models to support forecasts in C-LCA consists of the cut-off of some of the consequences to develop a decision-tree based on a set of assumptions linked to market, scale or temporal delimitations and, thereafter, their related effects (Schmidt 2008; Weidema et al. 2009). This modelling perspective is mainly based on the consideration of a number of economic variables and concepts, forming a coherent set of rules of thumb, and requires a deep knowledge of the markets affected by the decision investigated (Schmidt 2008). Contrary to the economic models, this approach assumes long-term full market elasticity and does not include indirect effects related to constrained production factors (e.g. the effects in the supply chain of the switch, for one production factor, from the supply to a client to another) but only indirect effects related to additional or reduced availability of co-products from multi-functional processes. A critical discussion on these modelling principles with respect to the main conceptual elements and variables of C-LCA is provided in Marvuglia et al. (2013).

2.3 Description of scenarios

In the current case study, a preliminary estimation of potential changes due to the predicted maize *shock* was developed using the GTAP model following the procedure established in previous studies (Kløverpris et al. 2008, 2010). Simulations were, therefore, run for Luxembourg[1] based on the FAO trade data (FAOSTAT 2012). These data indicated that Luxembourg is trading agricultural products overwhelmingly with its neighbouring countries: The Netherlands, Belgium, France and Germany. However, after running several *shocks* on the Luxembourgish agricultural sector, results showed that the impact on (national and foreign) production of a domestic biofuel policy would be very low, needing a shock of at least +100 % to begin to observe any changes in trade partner nations (Thomas Dandres, personal communication, CIRAIG, March 2012). Hence, results from the GTAP model suggested that there is no need for intensification or expansion beyond the national borders to meet the maize demand, due to the small economic impacts of agriculture in Luxembourg. In addition, these simulations also inhibit the usefulness of GTAP for this regionalised case study. The need to use more detailed models was therefore considered.

A first important remark has to be done on the peculiar nature of Luxembourgish farms, which are all integrated farms, practicing both crops cultivation and animal breeding at the same time. This important feature has driven the implementation of two different PE models to consider

[1] GTAP always considers data for Luxembourg coupled with Belgium, which constitutes a strong limitation in terms of the detail of the model for the specific needs of Luxembourg.

market constraints and the reaction of the agricultural system to the demanded production of maize at a given time horizon. The PE model used in *Approach A*, a linear programming model which is described in more detail in Rege et al. (2013) (see also section S2 of the supplementary material), was developed to seek the maximisation of revenues by farmers based on their crop cultivation and livestock activities. Based on the inventory data obtained from several data sources, the model was run to obtain the changes in crop production patterns in Luxembourg in the period between 2009 and 2020.[2] The change is however assessed in one single step, without showing the path of convergence leading from the 2009 situation to the 2020 pattern.

Approach B, while using the same PE model approach, and inspired on the model presented by Panichelli and Gnansounou (2008), aims at minimising a total opportunity cost[3] for satisfying a given demand of domestically produced bioenergy. Therefore, each activity is described by an opportunity cost, an upper limit of usage and a set of technical coefficients corresponding to the use of scarce resources linked to this activity. The different scenarios that were modelled with this approach, and presented in Table 1, allow the introduction of constraints on the number of displaced hectares of crops, on the availability of resources or on herd's size and animal feed.

Two important constraints in both methods are linked to the market boundaries and time horizon. In the former, both can be considered *single-country agricultural* models (Rege et al. 2013). Hence, all effects on new crop trade with foreign nations are still subject to scenario-based expert opinions. In the latter, both models show a static approach, lacking a convergence path through the length of the specific time window that has been applied (Rege et al. 2013). Nevertheless, it is important to note that there are some substantial differences between the two approaches that will be analysed in the discussion section.

Finally, a third perspective, named *Approach C*, was based on the decision-tree model described by Schmidt (2008), which focuses directly on creating a *consequential system delimitation for agricultural LCA* to determine which processes are marginally affected by a change in the main production system. This approach, therefore, seeks the modelling of the market influences on the analysed production system without combining LCA with external economic or modelling methodologies, in what could be defined an *endogenous* C-LCA modelling. Nevertheless, market information is still needed in this approach, by following the protocol defined by Weidema et al. (2009) and synthesised by Zamagni et al. (2012) for identifying the affected processes in C-LCA. *Approach C*, hence, was built upon the expert opinion of two agricultural economics specialists in Luxembourg. The two different scenarios that were modelled are described in Table 1 and in sections S3 and S4 of the Electronic supplementary material.

2.4 Function and functional unit selection

For this particular case study, the initial FU linked to the production of 80,000 tonnes was rescaled to the production of 1 MJ injected into the natural gas grid, as described by Jury et al. (2010), to allow direct comparability with conventional (i.e. fossil fuel-based) energy sources. The rationale for this rescaling is that the target audience associated with the research question behind this particular C-LCA case study was limited to policy-makers. This decision was taken based on the deliberation performed in Vázquez-Rowe et al. (2013), in which a distinction between *bounded* and *extended* indirect LUCs is made depending on the function of the system.[4] Indeed, if we assume that depending on the question that is being answered in a particular C-LCA the delimitations of the affected markets may change (Weidema et al. 2009), the consequences within the range of interest of policy-makers will be higher than for farmers, due to the fact that their decisions should be based on more complex market models. In other words, the interest of farmers is bounded to the agricultural sector, while policy-makers have to extend the scope to the wider energy and trade sectors in which agriculture can be fitted to understand the usefulness of applying such a policy. Hence, we considered that the complexity of a policy-making perspective would provide more robust results and discussion for the objectives of the examined study.

2.5 System boundaries

System boundaries were extended beyond the domestic agricultural system to account for the different consequences in terms of new import and export flows of agricultural products, constituting, as mentioned in "Section 2.4," an extended

[2] The distribution of cropland in Luxembourg in 2009 is considered the baseline scenario to model all three approaches (STATEC 2013).

[3] The additional economic effort implied by the choice to grow the second best (i.e. second most remunerative) crop available in terms of crop to plant.

[4] *Bounded* indirect LUCs refer to those changes in land use limited to the environmental burdens related to farmers' decisions of improving their economic revenue based on current trends in terms of crop prices and harvest yields. This decision context limits the system boundaries to the direct and indirect LUCs which occur within the Luxembourgish agricultural system. *Extended* indirect LUCs, in contrast, provide environmentally significant results to support policy-makers in deciding the appropriateness of enhancing bioenergy production. Hence, the system boundaries are extended to account for the different consequences in terms of new import and export flows of agricultural products, as well as including the entire chain of bioenergy production from maize.

Table 1 List of scenarios modelled in the three different approaches

Scenario acronym	Description
Approach A: PE model–revenue maximisation	
A1	LUCs are calculated with the computation of livestock opportunity costs. Feed supply, the dry matter, energy and protein content of the feed, the metabolic needs of the animals and the feed demand per animal were included.
A2	Minimal animal requirement was fixed at base level in these scenarios in order to detect how a change in the feeding patterns of the cattle would affect the agricultural system and the LUCs.
Approach B: PE model–opportunity cost minimisation	
B1	Optimal replacement and intensification for all crops.
B2	Optimal replacement and intensification, when only the additional *shock* is intensified.
B3	Optimal replacement and intensification when all the maize production may be intensified.
B4	Optimal replacement and intensification assuming that fertiliser use will only increase by 5 %.
B5	Optimal replacement and intensification of all crops and maize production under fertiliser constraint.
B6	Optimal replacement and intensification when herd sizes are reduced.
Approach C: consequential system delimitation for agricultural LCA approach	
C1	Expert opinion A. Considers intercontinental exchange to supply the Luxembourgish market.
C2	Expert opinions B and C. Considers buffering from neighbouring regions to supply the market.

approach. It should be noted that the three different approaches showed substantial differences in terms of system boundaries, as observed in Fig. 1 (see also Fig. S4 in the Electronic supplementary material), due to the differing expected consequences inside and outside the Luxembourgish boundaries. However, while the domestic variations are linked mainly to the inclusion of different crop types that may be affected by the maize *shock*, external boundaries varied significantly between approaches. The entire chain of bioenergy production from maize was also included.

Finally, it is important to mention that, despite the exclusion of livestock within the system boundaries of all three approaches, *Approach A* considers the interaction between crop production and the livestock sector in the PE model. This rationale is linked to the interest of farmers in obtaining low economic value feed to maximise their revenue and allows deepening the understanding of the complex interactions ongoing within the agricultural and livestock sectors themselves (Rege et al. 2013).

2.6 Life cycle inventory and life cycle impact assessment methods

The life cycle inventory (LCI) stage suffered small variations depending on the economic modelling of LUCs. Firstly, *Approach A* is fed by the LUCs calculated for the maximisation of revenues by the PE model. Therefore, all the domestic crops that were subject to changes in the model were included in the LCI. Additionally, all new import and export flows in the agricultural system, as described in "Section 2.3," were also modelled (Tables S7 and S8 in the Electronic supplementary material). The other two approaches, *B* and *C*, followed the same scheme with regard to the inventory items that were included, which translated into important differences in terms of the specific inventory items between the three approaches (see Table S8 in the Electronic supplementary material). Finally, the conversion step of maize production into biomethane and its purification and further consumption for the national grid was modelled based on the data available from Jury et al. (2010). All background processes were based on the ecoinvent® database (Frischknecht et al. 2007).

Regarding the life cycle impact assessment stage, the assessment method used for the computation of the LCA results were ReCiPe Midpoint and Endpoint (H) (Goedkoop et al. 2009). No further assessment methods were used, given that Vázquez-Rowe et al. (2013) determined that there were no major differences to be observed in the results for the Luxembourgish agricultural sector between methods.

3 Results

3.1 Predicted land use changes for the assessed approaches

Changes in land use patterns in arable land in Luxembourg for the three different approaches can be observed in Table 2. Approaches *A* and *B* present more complex shifts in land use changes (LUCs), linked to the optimisation performed by the PE models, while *Approach C* has limited consequential impact on changes in crop patterns since the expert opinion decision tree constitutes a simplification of the on going LUCs. Hence, the comparative analysis of the approaches shows the high level of discrepancy (and thus uncertainty) that can occur depending on the selected methods, regardless of the fact that they were elaborated taking into account the same methodological assumptions.

3.2 Environmental consequences with the PE model based on revenue maximisation—Approach A

Two different scenarios were selected for *Approach A*, which considers a PE model based on revenue maximisation

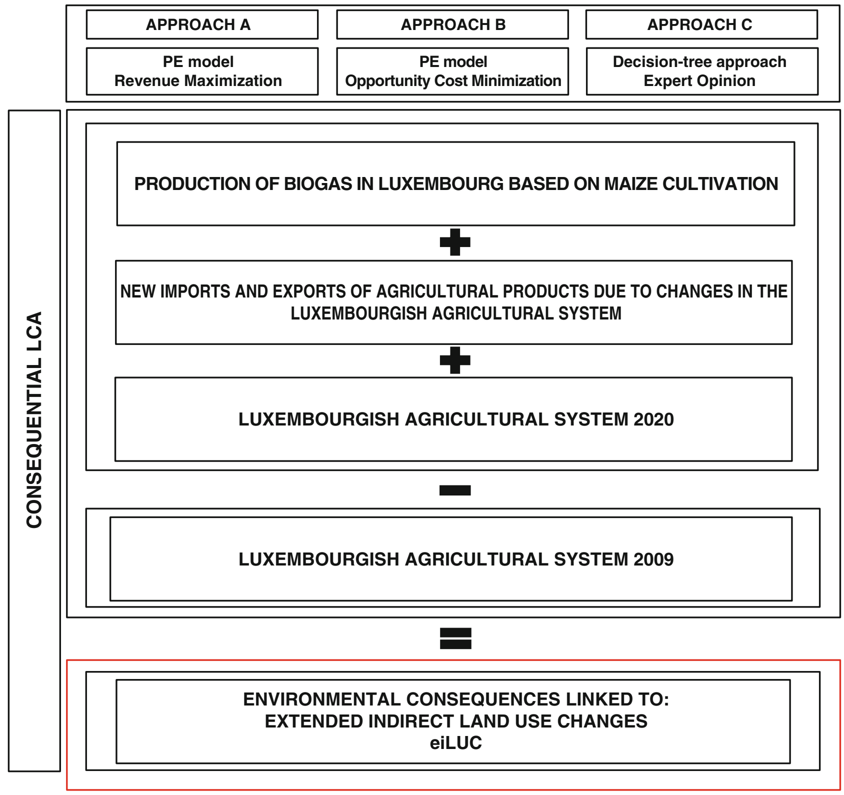

Fig. 1 Schematic representation of the included system boundaries of consequential life cycle inventory (LCI) modelling

Table 2 Changes in domestic arable land patterns per crop per model in the period 2009–2020

Crop	Unit	Approach A		Approach B						Approach C	
		A1	A2	B1	B2	B3	B4	B5	B6	C1	C2
Wheat spring	ha	−94.4	−94.4	−16.70	−41.79	−29.28	−31.53	−16.07	−91.93	−137.4	−200
Wheat winter (breadmaking)	ha	+2,432.8	+1,386.7	−223.91	−582.16	−407.80	−439.24	−223.91	−1280.6	0	−3,000
Wheat winter (fodder)	ha	−1,278.8	−1,278.8	−217.75	−566.15	−396.58	−427.16	−217.75	−1,245.4	−1,862.6	−500
Spelt	ha	−80	−80	−80	−80	−80	−80	−80	−80	−100	−100
Rye (breadmaking)	ha	+129.9	−70.2	−70.14	−31.88	−70.14	−70.14	−70.14	−70.14	0	0
Rye (fodder)		+277.5	−150	−149.86	−68.12	−149.86	−149.86	−149.86	−149.86	0	0
Barley winter (brewing)	ha	+1050.8	−72.8	−72.71	−33.06	−72.71	−51.45	−72.71	−72.71	−124.2	−100
Barley winter (fodder)	ha	+69.6	−1,099.8	−1,099.9	−500.04	−1,099.9	−778.23	−1,099.9	−1,099.9	−1,875.8	−100
Barley spring (brewing)	ha	+248.6	−134.4	−134.37	−87.62	−134.37	−95.07	−134.37	−134.37	−383.2	−100
Barley spring (fodder)	ha	+1,049.0	−567.0	−567.07	−257.82	−567.07	−401.25	−567.07	−567-07	−1,616.8	−100
Oats	ha	−276.8	−276.8	−277	0	−277	−126	−277	−277	−200	−200
Mixed grain	ha	−48.4	−48.4	−48	−48	−48	−48	−48	−48	0	0
Grain maize	ha	−81.8	−81.8	0	−37	0	0	0	−82	0	0
Triticale	ha	+1,500.4	−811	−811	−728	−811	−811	−811	−811	−700	−700
Forage crops	ha	−1,596.2	−1,596.2	−168	−188	0	−188	0	0	0	0
Rapeseed	ha	−925.8	−925.8	0	−408	0	−408	0	0	0	−3,000
Meadows and pastures	ha	−8,325	0	0	0	0	0	0	−59	0	0
Maize for bioenergy	ha	+5,949.2	+5,900.7	+3,856	+3,857	+4,242	+4,242	+3,856	+6,215	+8,000	+8,000
Maize intensification	t	0	0	27,261	27,261	21,987	21,987	27,261	0	0	0
Beans	ha	0	0	−15	−7	−15	−15	0	−15	0	0
Potatoes	ha	0	0	−61	−61	−61	−61	−61	−61	0	0

for farmers to estimate the LUCs. The endpoint results, shown in Fig. 2 (see also Table S9 in the Electronic supplementary material), forecast an increase in all environmental impacts for the two modelled scenarios. The considerable difference in burdens' increase when the two scenarios are matched (34 %) is linked mainly to a higher biogenic CO_2 retention due to LUCs and to lower agricultural land occupation (ALO) impacts in scenario A2. Nevertheless, the main carrier of impacts in both scenarios was fossil depletion (FD), linked to the transformation of maize for biomethane production.

3.3 Environmental consequences with the PE model based on opportunity cost minimisation—Approach B

The six different scenarios modelled within *Approach B* show a similar pattern in terms of environmental impact changes as compared with the baseline scenario (Fig. 3). In fact, if the worst-performing scenario (B1) is matched to the best-performing one (B3), environmental impact changes only differ by 21 %. Moreover, if this ratio is scaled-up to the total environmental impact generated by each scenario, the difference between the two is below 3 %. Furthermore, the magnitude of environmental impact changes (increases in all cases) is comparable to that identified for scenario A1 but substantially higher when compared with scenario A2 (up to 45 % higher). However, when the main carriers of the total environmental impact are analysed between the two approaches, there is a considerable distinction in terms of those impact categories that contribute the most. Therefore, while in *Approach A* the main impact categories, with contributions above 10 %, are FD, ALO and the two climate change (CC) categories, in *Approach B* FD and CC [HH] (climate change–human health) accounted for over 80 % of the total contribution to environmental consequences in all scenarios.

3.4 Environmental consequences with cut-off perspective—Approach C

Finally, two different models were based on the opinions of the consulted experts. Scenario C1, which considered an important supply of animal feed arriving from soybean production in South America, shows an overwhelmingly high contribution to the overall environmental impact of natural land transformation (NLT) impacts due to transformation of forest land in Brazil (Fig. 4). However, marine freighting of these products did not imply major environmental changes in terms of CC or FD. On the contrary, scenario C2 shows similar global environmental consequences to those obtained in Approaches *A* and *B*, with a clear dominance of CC and FD as carriers of environmental consequences, linked mainly to the higher energy intensity of expanding crops within Luxembourgish territory as compared with crops such as barley or wheat. Additionally, scenario C2 considers an important decrease in rapeseed production in Luxembourg (circa 3,000 ha). In the case of rapeseed, however, it was assumed that only 50 % of the reduction in production will eventually be imported from neighbouring regions, since this crop is used for industrial purposes outside Luxembourg and does not revert directly to the agricultural system in terms of animal feed or human consumption. Therefore, it was assumed that rapeseed would just be imported from other nations instead of from Luxembourg and, therefore, would not enter the domestic agricultural system in any case.

4 Discussion

4.1 Interpretation of main results

Whenever the LCA results are compared between the different scenarios, discrepancies due to LUCs are visible.

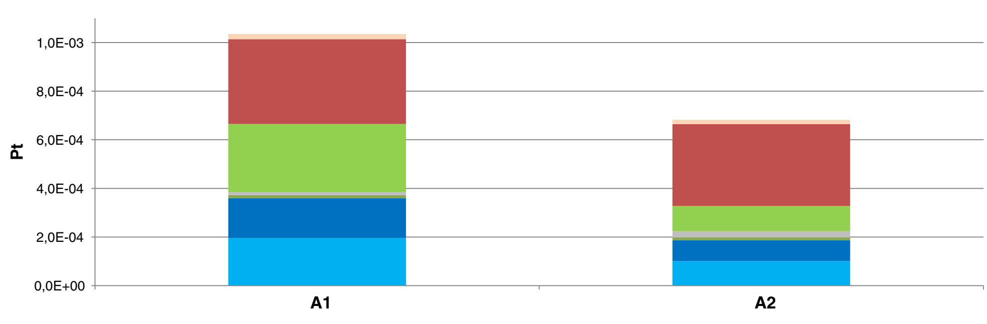

Fig. 2 Variation of single score endpoint values for the selected scenarios in *Approach A* as compared with the baseline scenario. Data per re-scaled FU=1 MJ. *Pt* endpoint single score points; *CC [HH]* climate change–human health; *POF* photochemical oxidant formation; *PMF* particulate matter formation; *CC [Ec]* climate change–ecosystems; *ALO* agricultural land occupation; *FD* fossil depletion; *Other*= other include the following impact categories: *OD* ozone depletion; *HT* human toxicity; *IR* ionising radiation; *TA* terrestrial acidification; *FE* freshwater eutrophication; *TET* terrestrial eco-toxicity; *FET* freshwater eco-toxicity; *MET* marine eco-toxicity; *ULO* urban land occupation; *NLT* natural land transformation; *MD* metal depletion

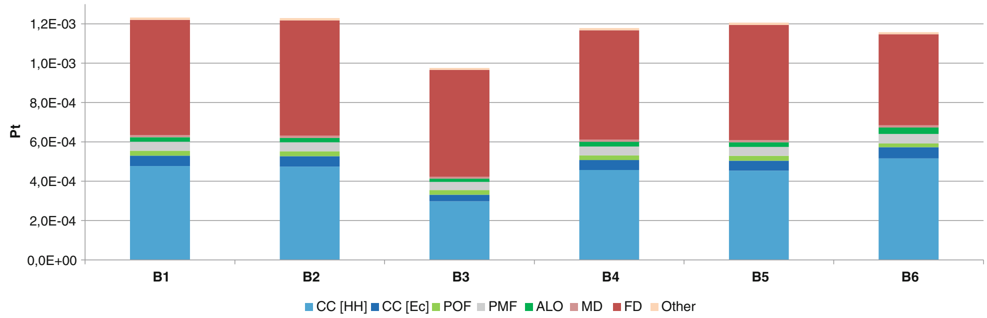

Fig. 3 Variation of single score endpoint values for the selected scenarios in *Approach B* as compared with the baseline scenario. Data per re-scaled FU=1 MJ. *Pt* endpoint single score points; *CC [HH]* climate change–human health; *POF* photochemical oxidant formation; *PMF* particulate matter formation; *CC [Ec]* climate change–ecosystems; *ALO* agricultural land occupation; *MD* metal depletion; *FD* fossil depletion; *Other*=other include the following impact categories: *OD* ozone depletion; *HT* human toxicity; *IR* ionising radiation; *TA* terrestrial acidification; *FE* freshwater eutrophication; *TET* terrestrial eco-toxicity; *FET* freshwater eco-toxicity; *MET* marine eco-toxicity; *ULO* urban land occupation; *NLT* natural land transformation

However, these disparities are also influenced by the environmental consequences occurring due to shifts in the import/export exchanges in Luxembourg due to LUCs. For instance, while approaches *A* and *B* show a relatively similar final single score value, due to the resembling trade-offs in terms of imports/exports and LUCs, *Approach C* shows much higher environmental impacts due to land use transformation impacts linked to soybean production in South America.

However, it should be noted that the variation between the different scenarios, as well as a relevant amount of the total changes in environmental impact, can be linked to the endogenous assumptions taken in each model, i.e. the underlying modelling structures that were created for each approach (Fig. 5), rather than the exogenous assumptions (i.e. the assumptions that were developed in the areas within the system boundaries that are not directly influenced by the modelling approaches, e.g. new import and export flows due to domestic LUCs). This is mainly due to the limited impact of the *shock* within the global trade market, which is also the reason why CGE models are not effective in this context, since they do not account for the high level of inventory detail that is necessary. However, it is important to remark that for scenario C1 the situation is considerably different, due to the expert opinions that predict a lack of buffer effect of surrounding crop land in neighbouring regions, under the assumption that importing soybean from South America would still be a more cost-effective measure.

Hence, for scenario C1, the environmental impact consequences are also highly attributable to the LUCs ongoing in South America concerning soybean production. Unlike in countries like Luxembourg or other European regions and nations, where the major changes in land use types go back decades and nowadays there is no margin of change in this respect, LUCs in soybean producing countries are not limited to crop rotation but may entail deforestation.

4.2 Advantages and limitations of the implemented approaches

As stated by Zamagni et al. (2012), the increased complexities, as well as uncertainties, constitute an integral limitation when interpreting the results derived from C-LCA studies.

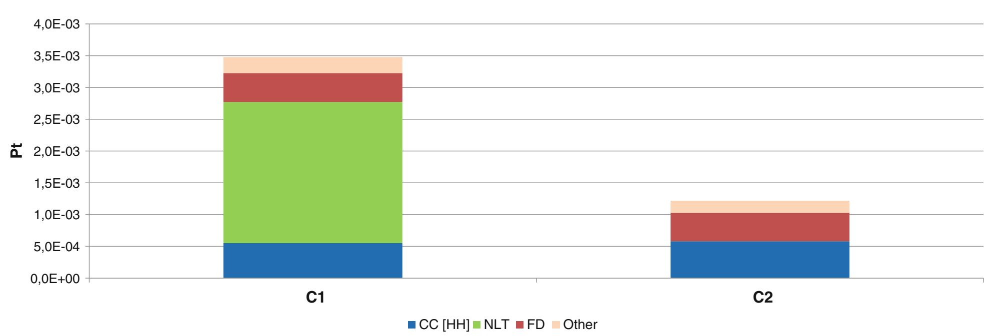

Fig. 4 Variation of single score endpoint values for the selected scenarios in *Approach C* as compared with the baseline scenario. Data per re-scaled FU=1 MJ. *Pt* endpoint single score points; *CC [HH]* climate change–human health; *NLT* natural land transformation; *FD* fossil depletion; *Other*=other include the following impact categories: ozone depletion; human toxicity; photochemical oxidant formation; particulate matter formation; ionising radiation; climate change–ecosystems; terrestrial acidification; freshwater eutrophication; terrestrial eco-toxicity; freshwater eco-toxicity; marine eco-toxicity; agricultural land occupation; urban land occupation; and metal depletion

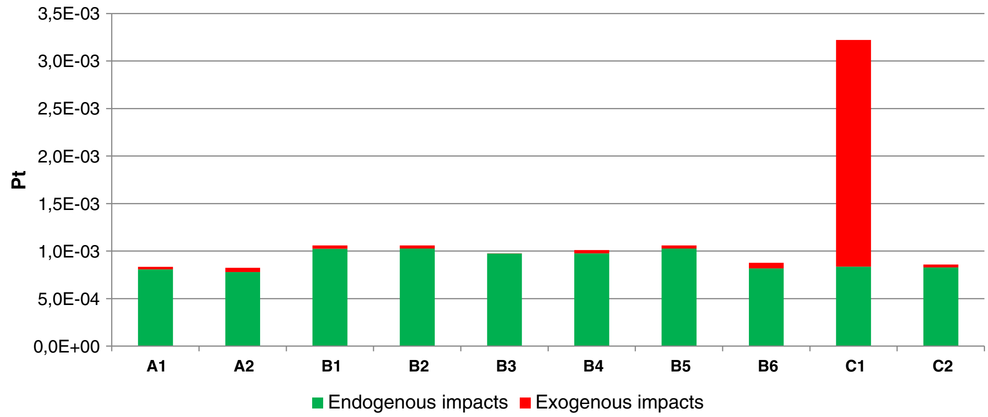

Fig. 5 Environmental impact consequences for the selected scenarios based on their exogenous or endogenous link to the economic models (results reported per re-scaled FU). Description for the different scenarios can be seen in Table 2; *Pt* endpoint single score points

However, these two key constraints are compensated by a wider scope of analysis of the assessed system, which provides a more comprehensive knowledge on the expected evolution of a specific production system following a change (Lesage et al. 2007; Schmidt 2010), which could not be correctly investigated through a conventional (attributional) approach.

More specifically, relating to the three C-LCA models presented in this study, a set of general issues can be highlighted regarding their comprehensiveness when giving feedback to the posed research question. In the first place, the two PE models, although fairly different in terms of internal structure and assumptions, both face a common problem, which is also linked to other models of this nature. Hence, PE models fail to inter-relate markets in a consistent way, providing a limited representation of reality and, therefore, constituting one of the major sources of uncertainty. Having said this, the fact that the two models are based on different approaches deserves some discussion. On the one hand, the PE model in *Approach A* considers several agricultural parameters (such as the crops' response to fertiliser use, endogenously), which implies the inclusion of non-linearity mechanism in the model. On the other hand, the model in *Approach B* only considers exogenous decisions linked to agricultural systems, and no optimisation in the decision-making process is foreseen.

Despite the fact that both approaches meet the main requirement of providing a market-linked model of the agricultural system in Luxembourg, *Approach B*, based on opportunity cost, only reflects marginal (small) changes for the different crops (see Table 2), under the assumption that individual crop cultivated areas will not influence in a determining way the overall market situation (Weidema et al. 2009). In contrast, the model in *Approach A*, as discussed in Rege et al. (2013), allows the implementation of large-scale effects within the individual crops. In fact, the 20 % *regulatory constraint* included in the model only responds to the legislative limitations in Luxembourg, which fix this value for annual crop rotation allowance (MAVDR 2005). Consequently, from a market-driven situation, it can be argued that *Approach A* depicts a series of forecast scenarios that resemble more closely the legislative and maximisation of revenues situation of Luxembourg's agricultural system.

Another constraint that is common to both approaches is the time horizon setting. Neither model accounts for the dynamic changes that may occur in the selected period (i.e. 2009–2020), which can constitute a major constraint in terms of price fluctuation in a very voluble market. In addition, the midterm changes that are monitored in the three approaches do not consider the inclusion of new technologies in the production system, which also introduces a considerable level of uncertainty in the results. Hence, the analysed scenarios should be seen as worst-case scenarios in terms of technological development, presuming that *technological constraints* in the evaluated production system will be relaxed through time (Weidema et al. 1999).

Finally, *Approach C* implies a higher level of perception and, therefore, uncertainty, due to the direct opinion of experts. This issue is enlarged in a small scientific community like the one existing in Luxembourg, where a brainstorming session with larger groups of experts was not feasible in order to complement private interviews. While individual expert opinions have to be seen as an important source of data, information and interpretation of a specific case study, brainstorming between experts may provide additional insights and interactions, since they probably will have different backgrounds that will allow fulfilling a wider range of knowledge gaps (Medsker et al. 1995; DeTombe 1999). However, despite the expected large uncertainties linked to the subjective estimations, data gaps in the LCI are an endemic limitation in life cycle thinking (Weidema and Wesnaes 1996; Reap et al. 2008; Finnveden et al. 2009) but usually become more complex in C-LCA due to the complex economic interrelations that are modelled (Mattsson et al. 2003; Brander et al. 2009; Sánchez et al. 2012).

4.3 Expert assumptions versus methodological modelling

Despite that the use of expert opinions to create a decision-tree on which to base consequential modelling may seem an

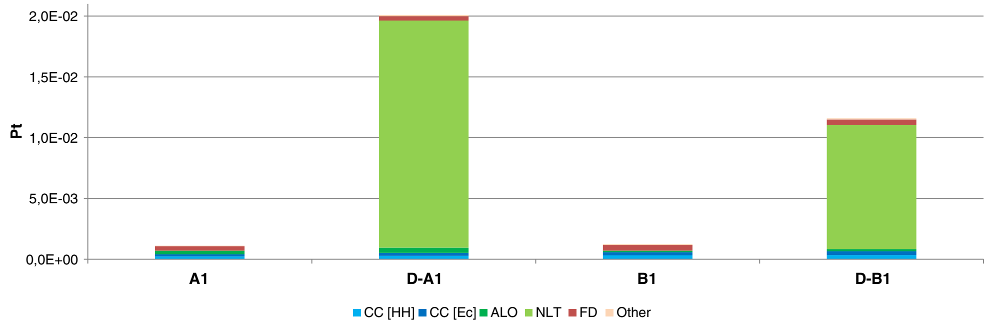

Fig. 6 Variation of single score endpoint values for selected scenarios integrating PE models with the expert assumptions modelling. Data per re-scaled FU=1 MJ. *Pt* endpoint single score points; *CC [HH]* climate change–human health; *CC [Ec]* climate change–ecosystems; *ALO* agricultural land occupation; *NLT* natural land transformation; *FD* fossil depletion; *Other*=ozone depletion; human toxicity; photochemical oxidant formation; particulate matter formation; ionising radiation; terrestrial acidification; freshwater eutrophication; terrestrial eco-toxicity; freshwater eco-toxicity; marine eco-toxicity; urban land occupation; and metal depletion

important source of uncertainty, it is important to take into consideration the fact that the quality of data in many production systems around the world is so limited that the use of expert opinions may be the only feasible possibility to conduct a study of these systems. Furthermore, all models have limitations. For instance, PE models fail to provide comprehensiveness when assessing the interactions between interrelated markets while CGE models (e.g. GTAP) have limited utility when analysing smaller-scale systems (as described in "Section 2.3").

Therefore, we argue that the complexity of production systems in reality, including their interrelations with markets and other production systems, cannot be fully encompassed through the inclusion of market models in consequential life cycle thinking (Ekvall and Andrae 2006; Schmidt 2010; Zamagni et al. 2012). In other words, despite the appropriateness of integrating economic models in C-LCA, these will always be somewhat constrained by their boundaries, leading practitioners to complement the modelling with assumptions, that, as stressed by Zamagni et al. (2012), may lead to significant errors when reporting final values (Reinhard and Zah 2009; Gaudreault et al. 2010; Smyth and Murphy 2011). Hence, the *consequential system delimitation for agricultural LCA*, which adapts the cut-off method proposed by Weidema (2003) to agricultural systems, can constitute an appropriate baseline consequential approach for all C-LCA studies, which should be substituted by adequate modelling mechanisms to reflect, when possible, the market boundaries that are affected by the changes analysed in the system. However, this substitution should be limited to those areas of the analysed production system which can effectively be monitored with these models and, therefore, maintain the cut-off method throughout the remaining market delimitations affected.

In the examined case study, it may be argued that the use of PE models is plainly justified due to the availability of data for the agricultural sector in Luxembourg. However, the models that are presented in this study are limited to the endogenous interrelations within the agricultural sector in Luxembourg, although different levels of detail can be distinguished (e.g. livestock inclusion in *Approach A*). Therefore, these models do not enable covering the entire interactions that define C-LCA, and expert opinions and other mechanisms should be included to aid those areas of the production system susceptible to high uncertainties. For instance, Fig. 6 shows the environmental consequences of two additional scenarios in which a hybrid modelling is performed combining the PE models (Approaches *A* and *B*) with the expert opinion method (*Approach C*) and comparing the results with those obtained throughout the manuscript, in which the modelling methods are assessed independently. This alternative approach allows deepening the consequential thinking by applying the different modelling approaches in an integrated manner in those areas within the system boundaries where they are most appropriate. In fact, as can be observed, there are substantial differences in environmental consequences when the two methods are combined, stressing the need to account for the highest possible portion of the system under analysis.

5 Conclusions

The results highlight the importance of methodological assumptions whenever developing C-LCA. For instance, the economic model selection and the delimitation of the consequences proved to be high sources of result variability. In fact, despite the advantages and limitations of all three consequential modelling approaches, none of them were able to fully supply an integrated approach to fulfil the entire cascade of relevant consequences identified in the case study. Hence, future development will focus on the interpretation of these differences and on possible pathways to harmonise these disparities.

Scenario analysis has an increasingly important role both in A-LCA and C-LCA. In the former, there is in fact an increasing demand to report a comprehensive analysis of

how different methodological scenarios affect the results of the assessed system (i.e. FU, allocation, system boundaries …); in the latter, in many C-LCA studies, scenario modelling is required to provide a wide set of potential forecasts. We argue that the use of several C-LCI modelling approaches, such as PE, CGE or cut-off models, within the same study may help widen the interpretation of the advantages or risks of applying a specific change to a production system. In fact, different models may not only be good alternatives in terms of comparability of scenarios and assumptions, but there may also be room for complementing these within a unique framework with the aim of reducing the uncertainties of the system in an integrated way.

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